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Modelling climate change impacts on nutrients and primary production in coastal waters

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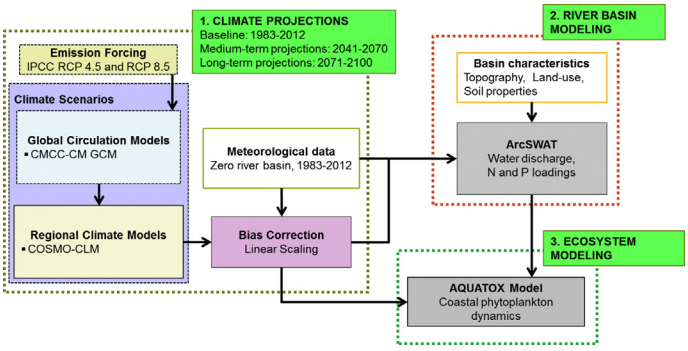
HIGHLIGHTS

- Integrated modelling methodology exploring the impacts of climate change on nutrient loadings and phytoplankton community.
- The nutrients loadings in the Zero river basin and the salt-marsh Palude di Cona in Italy were investigated.
- Integration of GCM-RCM climate projections with the hydrological model SWAT and the ecological model AQUATOX.
- Results show changes in the phytoplankton community: reduction in biodiversity and more intense blooms in the summer.

GRAPHICAL ABSTRACT



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ABSTRACT

There is high confidence that the anthropogenic increase of atmospheric greenhouse gases (GHGs) is causing modifications in the Earth's climate. Coastal waterbodies such as estuaries, bays and lagoons are among those most affected by the ongoing changes in climate. Being located at the land-sea interface, such waterbodies are subjected to the combined changes in the physical-chemical processes of atmosphere, upstream land and coastal waters. Particularly, climate change is expected to alter phytoplankton communities by changing their environmental drivers (especially climate-related), thus exacerbating the symptoms of eutrophication events, such as hypoxia, harmful algal blooms (HAB) and loss of habitat. A better understanding of the links between climate-related drivers and phytoplankton is therefore necessary for projecting climate change impacts on aquatic ecosystems.

Here we present the case study of the Zero river basin in Italy, one of the main contributors of freshwater and nutrient to the salt-marsh Palude di Cona, a coastal waterbody belonging to the lagoon of Venice. To project the impacts of climate change on freshwater inputs, nutrient loadings and their effects on the phytoplankton community of the receiving waterbody, we formulated and applied an integrated modelling approach made of: climate simulations derived by coupling a General Circulation Model (GCM) and a Regional Climate Model (RCM) under alternative emission scenarios, the hydrological model Soil and Water Assessment Tool (SWAT) and the ecological model AQUATOX. Climate projections point out an increase of precipitations in the winter period and a decrease in the summer months, while temperature shows a significant increase over the whole year. Water discharge and nutrient loads simulated by SWAT show a tendency to increase (decrease) in the winter

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(summer) period. AQUATOX projects changes in the concentration of nutrients in the salt-marsh Palude di Cona, and variations in the biomass and species of the phytoplankton community.

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1. Introduction

There is high confidence that anthropogenic emissions of greenhouse gases (GHGs) are the main reason of the current Earth's energy imbalance (Hansen and Sato, 2004). As a consequence, this is causing the warming of the atmosphere (von Schuckmann et al., 2016) with an expected rise of the global mean temperatures by 0.3 to 4.8 °C by the end of the 21st century relative to the period 1986–2005 (IPCC, 2013), and the alteration of the water cycle as a result of changes in the global moisture recycling among atmosphere, lands and oceans (Levang and Schmitt, 2015).

Coastal ecosystems, together with the ecological and socio-economic services they provide, could be among those most affected by the ongoing climate change (Harley et al., 2006; IPCC, 2014). Coastal waterbodies such as estuaries, bays and lagoons, being transitional systems located at the interface between land and sea, will be subjected to the combined modifications taking place in the atmosphere, in the oceans, and over the land surface (Raimonet and Cloern, 2016).

In particular, a wide number of studies has shown the links between changes in climate and phytoplankton in coastal ecosystems (Harding et al., 2015; Holt et al., 2016; Quigg et al., 2013; Sommer and Lewandowska, 2011; Winder and Sommer, 2012). Phytoplankton is responsible for a large share of photosynthesis and primary production of coastal areas, and plays an essential role in several biogeochemical processes such as carbon, nutrient and oxygen cycles (Paerl and Justic, 2011; Winder and Sommer, 2012). Consequently, changes in phytoplankton dynamics and composition may have relevant repercussions on the higher trophic level of coastal ecosystems (Hernandez-Farinas et al., 2014; Schloss et al., 2014).

Environmental drivers regulating phytoplankton dynamics such as water temperature, light penetration, tides, salinity and nutrient availability (Cloern, 1996), are highly sensitive to climate change. The interactions among these drivers, including their influence on primary production in coastal areas, form a complex, nonlinear system that can be defined by three main components: (1) Climate, which delineates changes in atmospheric conditions; (2) Coastal watershed hydrology, which describes the changes in the delivery of freshwater, nutrients, sediments and pollutants to coastal waters; and (3) coastal ecosystem, which combines changes in climate and watershed hydrology to identify the impacts on the coastal environments. As proposed by Xia et al. (2016) for freshwater ecosystems, the simplified conceptual model depicted in Fig. 1(a) describes the interactions among these components and their effects on the phytoplankton communities of coastal ecosystems.

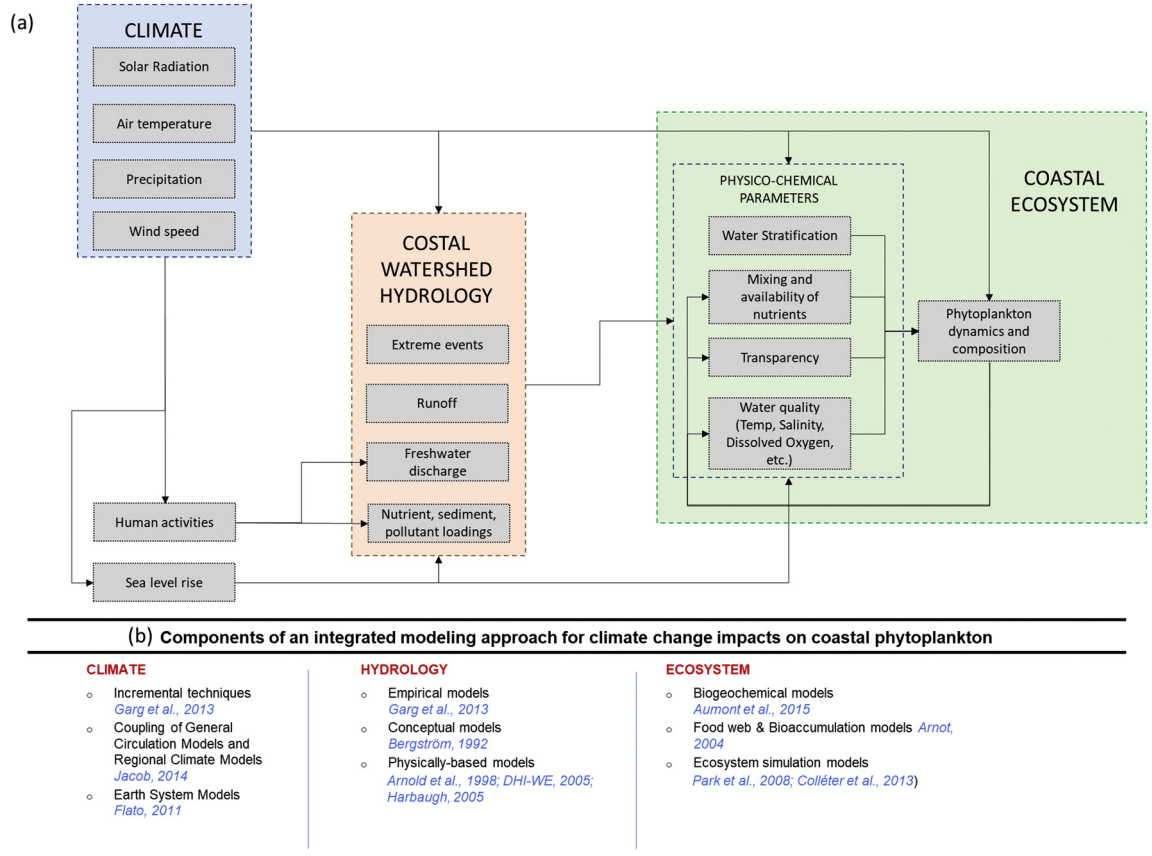

(a)
CLIMATE
Solar Radiation
Air temperature
Precipitation
Wind speed
COSTAL WATERSHED HYDROLOGY
Extreme events
Runoff
Freshwater discharge
Nutrient, sediment, pollutant loadings
Human activities
Sea level rise
COASTAL ECOSYSTEM
PHYSICO-CHEMICAL PARAMETERS
Water Stratification
Mixing and availability of nutrients
Transparency
Water quality (Temp, Salinity, Dissolved Oxygen, etc.)
Phytoplankton dynamics and composition
(b) Components of an integrated modeling approach for climate change impacts on coastal phytoplankton
CLIMATE
Incremental techniques Garg et al., 2013
Coupling of General Circulation Models and Regional Climate Models Jacob, 2014
Earth System Models Flato, 2011
HYDROLOGY
Empirical models Garg et al., 2013
Conceptual models Bergström, 1992
Physically-based models Arnold et al., 1998; DHI-WE, 2005; Harbaugh, 2005
ECOSYSTEM
Biogeochemical models Aumont et al., 2015
Food web & Bioaccumulation models Arnot, 2004
Ecosystem simulation models Park et al., 2008; Colléter et al., 2013)

Fig. 1. (a) Conceptual model describing the complex interactions among environmental and climate drivers regulating phytoplankton dynamics in coastal areas. The model is composed of three parts: (1) climate, (2) coastal watershed hydrology; and (3) coastal ecosystem. (b) Examples of models able to represent the three components (climate, watershed hydrology, ecosystem) and that can be integrated to constitute a modelling approach for the assessment of the direct and indirect impacts of climate change on phytoplankton (and higher trophic levels) of coastal ecosystems.

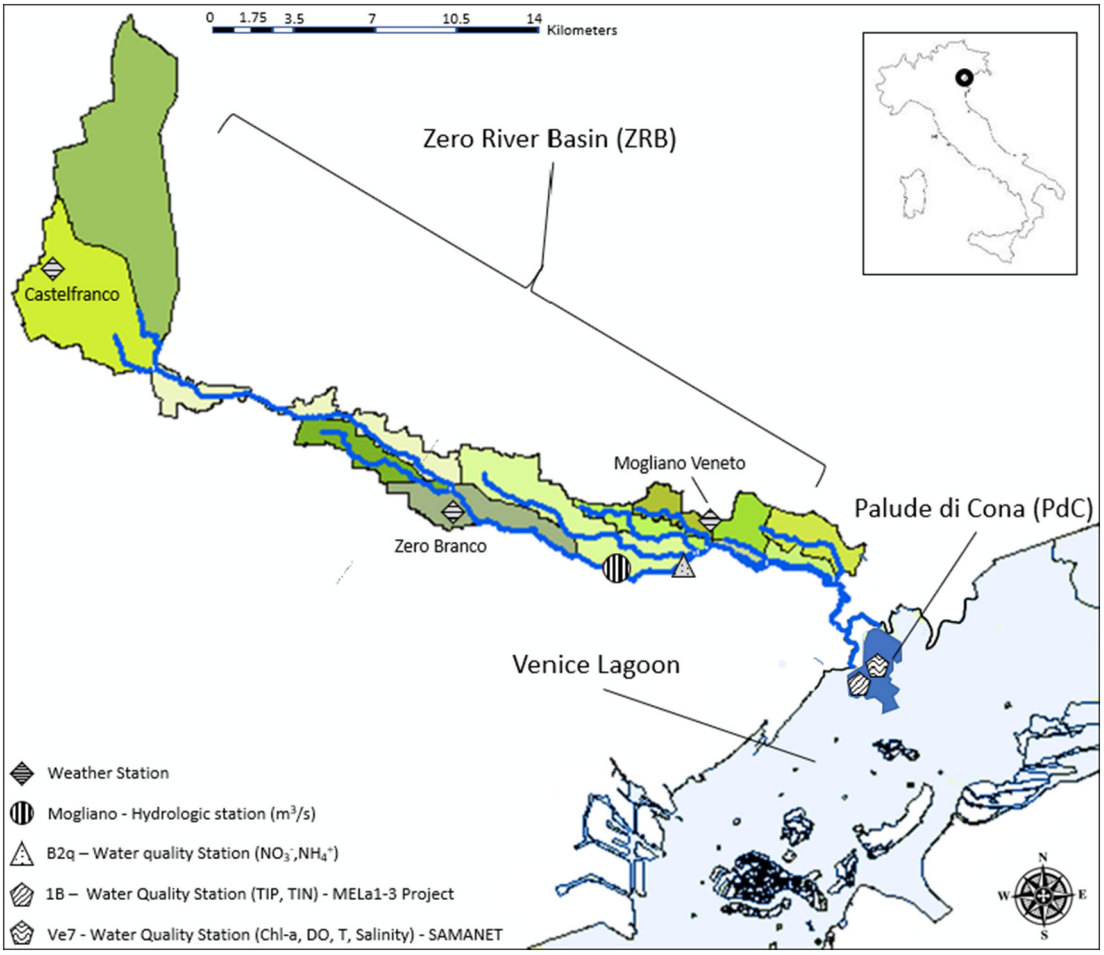


Fig. 2. Map representing the Zero river basin (ZRB) and its sub-basins, Palude di Cona (PdC) and the monitoring stations used in the study.

The conceptual model helps to illustrate the cascading mechanisms linked to climate change and leading to direct and indirect impacts on coastal phytoplankton. In particular, the distribution, abundance and structure of phytoplankton communities, as well as their phenology and productivity, are changing in response to warming and stratifying waters (Hunter-Cevera et al., 2016; Lassen et al., 2010; Weisse et al., 2016). Specifically, a prolonged thermal stratification affect phytoplankton sinking velocity and can advantage smaller and buoyant species of phytoplankton (Diehl et al., 2002; Huisman et al., 2004; Marinov et al., 2010). Moreover, a prolonged thermal stratification in the water column can suppress the upward flux of nutrients from deep layers, resulting in more frequent nutrient-depleted conditions at the surface, and further advantaging those species that are already winners in the competition for nutrients (Schmittner, 2005).

Variations in precipitation patterns and increased frequency of extreme events such as heavy rainfalls and droughts can alter those physical processes (e.g. runoff, leaching, percolation) that regulate the delivery of freshwater and its load of nutrients, sediment and pollutants to coastal waterbodies (Hagy et al., 2004; IPCC, 2014; Moss et al., 2011; Xia et al., 2016), with consequences on their nutrient and salinity concentrations (Dimberg and Bryhn, 2014). Decrease in precipitation could reduce river flow and both nutrient loadings and dilution (Whitehead et al., 2009). Increased winter precipitation and more frequent and severe flood events during summer will increase runoff and associated wash-off of nutrients and sediments (Jeppesen et al. 2009a, b; Najafi and Moradkhani, 2015; Sterk et al., 2016; Whitehead et al., 2009). These events may alter the nutrient budget and change the macronutrient stoichiometry (N:P:Si) of coastal waters (Statham, 2012; Winder and Sommer, 2012), resulting in more frequent eutrophication events and appearance of undesirable phytoplanktonic species such as cyanobacteria (Domingues et al., 2017; Garnier et al., 2010; Kaur-Kahlon et al., 2016).

Salinity of coastal waterbodies such as lagoons depends on the balance between marine and freshwater (either from surface of subsurface) inputs, often altered by climate-induced changes in the dynamics of runoff regimes and sea level rise (Fichez et al., 2017; Telesh and Khlebovich, 2010; Vargas et al., 2017) with consequences in structure, dominant species and biomass production of the plankton communities (Dhib et al., 2013; Flöder et al., 2010; Kaur-Kahlon et al., 2016; Olenina et al., 2016).

Sea level rise might have further important consequences on coastal phytoplankton, especially in shallow waters. For example, shallow lagoons can have well-developed benthic phytoplankton communities that can contribute to a major portion of the fixed carbon in the system (Parodi and De Cao, 2002). Increased water depth due to sea level rise will cause benthic phytoplankton to capture a smaller proportion of the solar radiation due to the stronger light attenuation in the water column (Brito et al., 2012).

Additionally, other anthropogenic disturbances (pollution, water withdrawal, etc.) may act in synergy with climate change and impact phytoplankton in coastal waters (Liu and Chan, 2016; Rabalais et al., 2009). For example, nutrient and pollutant loads are expected to increase in the coming decades due to population growth, increased use of inorganic fertilizers, manure and pesticides, increased fossil fuel burning (associated emission of NO_x), and expansion of sea-based activities such as aquaculture (Bouwman et al., 2009; Burkholder et al., 2007; Verdegem, 2013). On the other hand, factors such as increased production costs of fertilizers (Blanco, 2011), and promotion of good

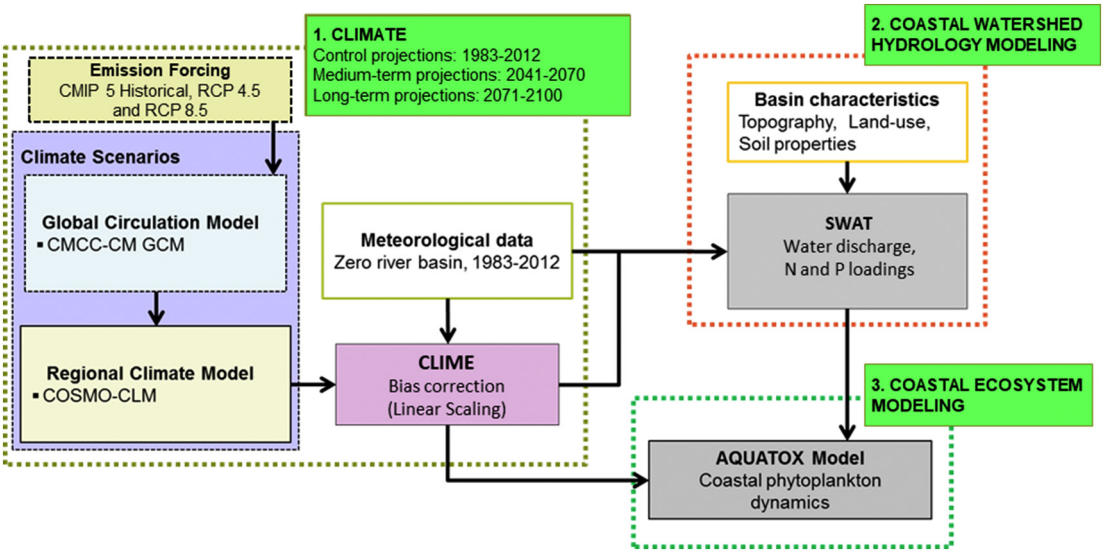


Fig. 3. Workflow of the Integrated Modelling Approach applied in the study.

agricultural practices (GAPs) and integrated nutrient management (INM) might balance out the increase fertilizer usage.

The described effects generated by climate change and other anthropogenic stressors, and their balancing and reinforcing interactions, are thus expected to change the dynamics and composition of coastal phytoplankton communities, and increase the frequency and abundance of eutrophication events and related symptoms such as hypoxia, harmful algal blooms (HAB), unsightly scums and loss of habitat (Lloret et al., 2008; Moore et al., 2008; O'Neil et al., 2012; Paerl and Paul, 2012). While being able to project these effects is of fundamental importance, the complexity of coastal systems makes however difficult and challenging to attribute changes in the abundance and composition of coastal phytoplankton to specific causes (Facca and Sfriso, 2009) (Statham, 2012).

In this view, the integration of climate simulations and mechanistic environmental models can become a valuable tool for the investigation and projections of phytoplankton dynamics under climate change. The integration of models has already been promoted as a tool able to support Integrated Coastal Zone Management (ICZM) and Integrated Water Resource Management (IWRM), as it allows to represent the complexity of such systems by facilitating the simulation and combination of multi-faceted drivers such as climate conditions, nutrient availability, water withdrawal, and other pressures (Brush and Harris, 2010; ComEC, 2000). Integrated modelling can help to better understand and reproduce the cause-effect relationships between different components of complex systems such as coastal waterbodies, and they have become necessary tools for managers and decision makers who have to consider impacts on different end-points (i.e. social, economic, environmental) (Jakeman and Letcher, 2003). Different types of models (Fig. 1(b)) can be adopted and integrated to study the interactions and feedbacks among the three components (climate, watershed hydrology, coastal ecosystem) of the conceptual model depicted in Fig. 1(a). In the last decades, the adoption of mechanistic models has become a popular tool for assessing the impacts of climate change on hydrological and abiotic components of aquatic systems (Vohland et al., 2014). However, the majority of studies have analyzed the impacts of climate change on single environmental aspects such as watershed hydrology and water availability (Amin et al., 2017; Leta et al., 2016; Trinh et al., 2017), loadings of nutrients (Huttunen et al., 2015; Piniewski et al., 2014) and sediments (Bussi et al., 2016; Samaras and Koutitas, 2014), and water quality (Nielsen et al., 2014; Wilby et al., 2006). Moreover, these studies are limited to the assessment of the impacts on abiotic components, and are not able to model the consequences on phytoplankton and higher trophic levels. Although the number of studies that attempt to integrate climate simulations and process-based models for assessing the impacts of climate change on primary producers and aquatic ecosystems is growing (Elliott, 2012; Glibert et al., 2014; Guse et al., 2015; Longo et al., 2016; Mooij et al., 2007; Taner et al., 2011; Trolle et al., 2015), with the merit of trying to address the impacts at the ecosystem level, this number is still very limited and mainly focused on specific types of water bodies, such as lakes (Mooij et al., 2010).

Table 1
Description of models and tools involved in the Integrated Modelling Approach. The columns "Input" and "Output" clarify the connection between the components in the presented application and focus on selected climate-related inputs and outputs of each model/tool.

Component	Model/Tool	Type of model/tool	Input	Output
Climate projections	General Circulation Model (GCM) and Regional Climate Model (RCM)	GCM/RCM nested simulations for the 20th and the 21st century	Historical forcing experiment from 1850 to 2005, RCP 4.5 and RCP 8.5 experiments from 2006 to 2100	Daily time series of climate variable (Precipitation and Temperature) (Climate models were not run in this study as data were already available from previous modelling studies)
Climate projections	CLIME (Villani et al., 2015)	GIS Tool for climate analysis (bias-correction and model evaluation)	Daily time series of climate variables (Precipitation and Temperature) from GCM/RCM nested simulations and from weather stations for the same historical period	Bias-corrected daily time series of climate variables (Precipitation and Temperature)
Watershed hydrology	SWAT (Arnold et al., 1998)	Semi-distributed hydrological model	Bias-corrected daily time series of climate variables (Precipitation and Temperature)	Monthly time series of water discharge and nutrient loads
Ecosystem modelling	AQUATOX (Park et al., 2008)	Ecological model for aquatic environments	Monthly time series of water discharge and nutrient loads by SWAT, and of water temperature calculated from regression with air temperature	Monthly time series of phytoplankton abundance (Chlorophyll-a) and composition

Table 2
Scenarios adopted in the study. Future mid-term (2041–2070) and long-term (2071–2100) simulated scenarios were compared with the control simulated period (1983–2012).

Emission scenario	Time frames	Years
Historical	Control	1983–2005
RCP4.5		2006–2012
RCP8.5		2006–2012
RCP4.5	Mid term	2041–2070
	Long term	2071–2100
RCP8.5	Mid term	2041–2070
	Long term	2071–2100

Therefore, there is need for additional studies able to provide new and improved methods to investigate the independent and combined effects of climate change and other anthropogenic stressors on aquatic ecosystems (Trolle et al., 2014). This is particularly true for coastal waterbodies. The inherent difficulties in modelling a cyclic system with interactions and feedbacks (both balancing and reinforcing) among land, sea, atmosphere and biota (Cossarini et al., 2008; Videira et al., 2011) are due to constraints such as lack of data and knowledge gaps about the spatial and temporal complexity of responses and of comprehensive feedback loops (Clark, 2001; Rode et al., 2010), as well as about delayed and off-site effects. The development and test of new integrated modelling approaches can provide a useful contribution to reinforce the reliability of management tools for ecological conservation and adaptation policies of sensitive aquatic ecosystems such as those in coastal areas.

In this perspective, here we present an integrated modelling approach for assessing potential medium (2041–2070) and long-term (2071–2100) effects of climate change, represented by changes in air temperature and precipitation, on the productivity and community structure of coastal phytoplankton at a local scale. This approach can deepen the knowledge about the interrelated impacts of climate change along the land-water continuum, from climate-related effects on stream flow and nutrient loadings, to direct influence of temperature changes on coastal waters. According to Fig. 1, this approach consists of coupled General Circulation Model/Regional Climate Model (GCM/RCM hereafter) nested simulations, able to provide localized climate data suitable for impact assessment studies, and two separate environmental models, the hydrological model Soil and Water Assessment Tool (SWAT; (Arnold et al., 1998) and the ecological model AQUATOX (Park et al., 2008) used to depict the physical, chemical and biological process of a coastal watershed and of the receiving waters in a coastal environment. This study is a further attempt to integrate climate projections and tools with environmental models for assessing the responses of aquatic ecosystems to climate change. The main objective of the paper is to illustrate the modelling approach, demonstrate its applicability through a local case study, and to discuss potential, limitations and areas of improvement.

Table 3
Most sensitive parameters identified in the simulation of the ZRB. Parameters regulate hydrologic processes (1–8), nitrogen processes (9–10), and phosphorus processes (11−12) and were used for the calibration of the model.

Parameter #	Symbol	Value
1	GW_DELAY	184.0
2	OV_N	9.3
3	SOL_BD	0.35
4	REVAPMN	208.68
5	GW_REVAP	0.23
6	ALPHA_BF	2
7	CN2	−0.25
8	HRU_SLP	0.06
9	ERORGN	8.8
10	HLIFE_NGW	63
11	BC4	0.5
12	RS5	0.1

GW_DELAY: groundwater delay; OV_N: Manning's "n" value for overland flow; SOL_BD: moist bulk density; REVAPMN: threshold depth of water in the shallow aquifer for "revap" to occur (mm); GW_REVAP: groundwater 'revap' coefficient; ALPHA_BF: baseflow alpha factor (days); CN2: SCS runoff curve number f; HRU_SLP: Average slope steepness; ERORGN: organic N enrichment ratio; HLIFE_NGW: half-life of nitrate in the shallow aquifer (days); BC4: rate constant for mineralization of organic P to dissolved P in the reach at 20 °C; RS5: organic phosphorus settling rate in the reach at 20 °C.

2. Material and methods

2.1. Study area

The Zero river basin (ZRB) and the waters of the salt-marsh Palude di Cona (PdC) belong to the land-water continuum of the lagoon of Venice, Italy (Fig. 2).

2.1.1. Zero river basin

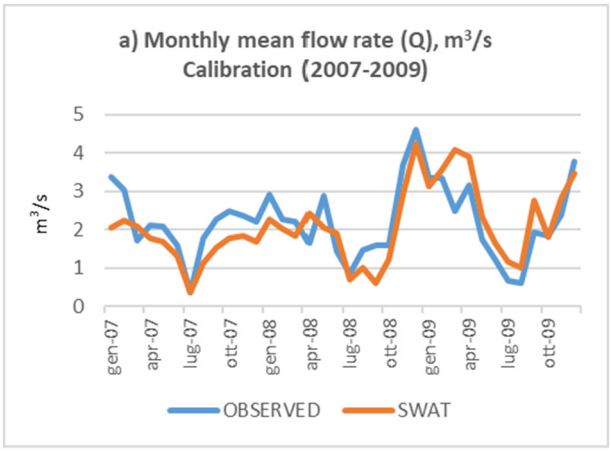
The ZRB has a surface of 140 km^2 and an elevation range from 1 to 110 m a.s.l. Agriculture is the dominant land use in the Zero catchment (72%) with corn, soy and wheat as dominant crops (ARPAV, 2009b). Urban and industrial areas (24%), and semi-natural and forested areas (4%) cover the remaining surface. The north-western part is characterized by a significant presence of livestock farms, with a density of 5 to 10 farms per km^2 (ARPAV, 2001). Agricultural activities provide the greatest contribution of chemical inputs, specifically, synthetic fertilizers and organic fertilizers in the form of manure and urea. The area features a Mediterranean climate with peculiarities typical of the neighboring more Continental climates (Guerzoni and Tagliapietra, 2006): mild winters and dry summers, common of Mediterranean climates, are replaced by cold winters and summers with frequent storms. Annual average temperature is 14 °C, with January and December being the coldest months (around 4 °C), and July and August the warmest (around 25 °C). The average annual rainfall is 1000 mm, with peaks in spring and autumn and minimums in winter and summer periods. Different external factors influence the hydrology and nutrient loads of the Zero river (Essenfelder et al., 2016), in particular the numerous hydraulic infrastructures that regulate the streamflow necessary to satisfy the different water needs in the area and the groundwater contributions coming from the external aquifer of the Venetian high plains (Servizio Acque Interne, 2008).

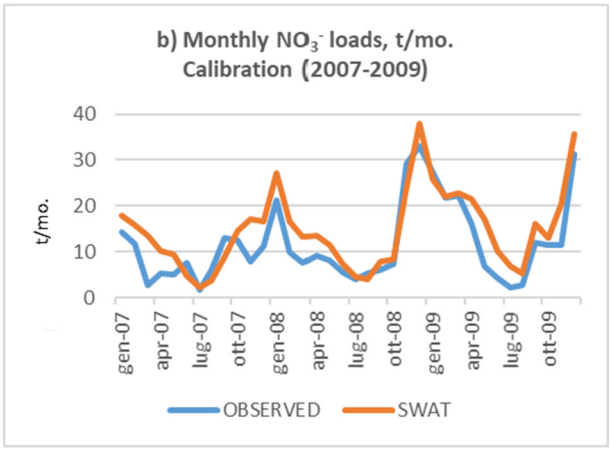
2.1.2. Palude di Cona

The PdC is a shallow water body surrounded by salt-marshes and located in the upper-north basin of the lagoon of Venice. It is 4 km long, 0.9 km to 1.7 km wide, with an average depth of 80 cm during mean tidal conditions of the microtidal cycle of the Venice lagoon (Sarretta et al., 2010). It is surrounded and crisscrossed by navigation channels affecting its hydrology. PdC, as any area of the Venice lagoon, is influenced by the effects of the tide, which exposes the bottom surface of the area during low tide events. From the meteorological observations for the

Table 4
Phytoplankton compartments added to the system.

N.	Species	Optimal T (°C)	N Half-sat K	P Half-sat K
D1	*Navicula* ssp.	15	0.01	0.002
D2	*Cyclotella nana*	20	0.011	0.017
D3	*Cyclotella nana* (high nutrient waters)	20	0.117	0.055
D4	*Fragilaria* spp. (low nutrient waters)	26	0.0154	0.001
D5	*Cyclotella nana* (warm waters)	25	0.011	0.017
D6	Cyclotella *nana* (extremely warm waters)	30	0.011	0.017
D7	*Cyclotella* spp. (high nutrient and warm waters)	25	0.117	0.055
D8	*Fragilaria* spp. (high nutrient and cold waters)	8	0.117	0.05
CB1	*Microcystis* spp.	30	0.4	0.03





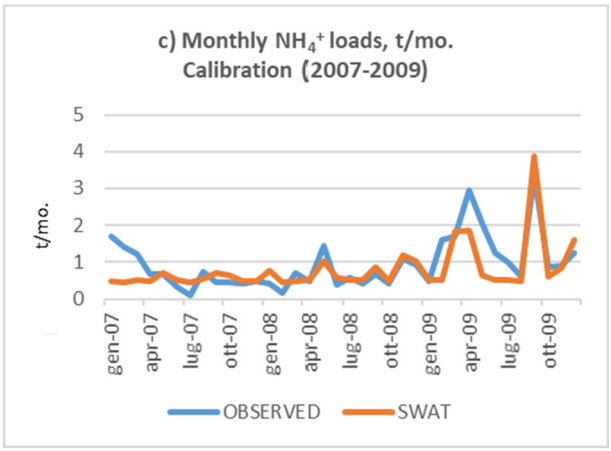
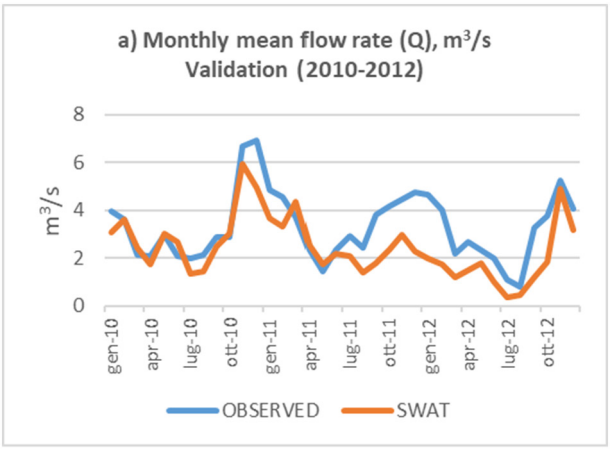
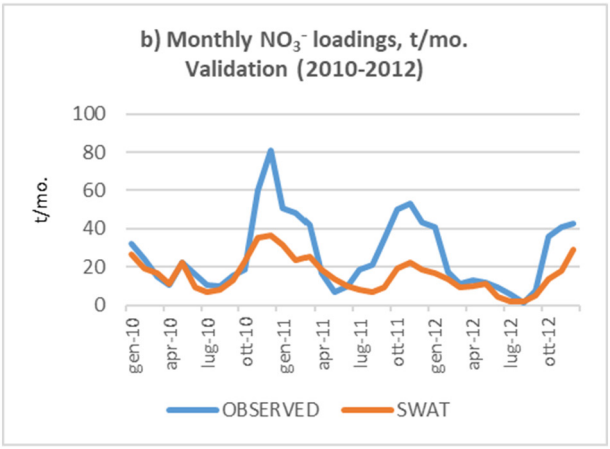


Fig. 4. Calibration results of the SWAT model: a) water discharge; b) $N\text{-}NO_3^-$; c) $N\text{-}NH_4^+$.





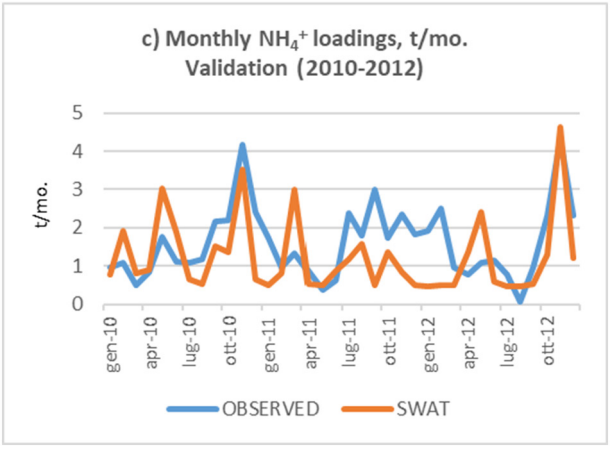


Fig. 5. Validation results of the SWAT model: a) water discharge; b) $N\text{-}NO_3^-$; c) $N\text{-}NH_4^+$.

period 2002–2012 (ARPAV, 2013a) temperatures are, on average, 0.5–1 °C warmer than those of the Zero river basin and annual precipitations are, on average, 200 to 300 mm lower (Guerzoni and Tagliapietra, 2006), while solar radiation, important factor influencing photosynthetic processes in primary producers, reaches the highest peaks of 25 MJ/m^2 day in summer and the lowest of 5 MJ/m^2 day in winter.

The shallow waters of PdC promote rapid temperature equilibrium between water and air, with water temperature following the seasonal trends of air temperature (Guerzoni and Tagliapietra, 2006). The water temperature in PdC shows the highest values in the summer, reaching 26–27 °C, while wintertime lows are typically around 5–7 °C, as indicated by monitoring data for the period 2007–2012 (SAMANET, 2013). Dissolved oxygen (DO) concentrations fluctuate with water temperature seasonally as well as diurnally. Monthly averages for the 2007–2012 period reach 10–12 mg/l in winter, while summertime lows are typically around 4–6 mg/l (SAMANET, 2013).

The trophic state of a transitional environment such as PdC is the result of multiple factors such as the loadings and concentrations of nutrients, freshwater discharge, tidal excursion, climate conditions and biological processes (Cloern, 2001). The nitrogen-phosphorus ratio (N:P ratio) in PdC assumes higher values than the Redfield ratio (16:1), commonly used to describe the composition of marine phytoplankton (Zirino et al., 2016a). Seasonal values of N:P ratios in Pdc for the 2007–2009 period (MAV, 2004) oscillates from values of 65:1–55:1 in winter to values of 27:1–24:1 in summer. Carstensen et al. (2011)

Table 5
Statistical comparison of means and variances between observed and modeled (SWAT) phosphorus values.

Period	Mean SWAT (kg)	Mean obs. (kg)	St. dev. obs.	Variance SWAT	Variance obs.	rB	F
2007–2009 (*calibration*)	498.49	646.52	264.07	51,496.89	69,730.59	−0.56	0.74
2010–2012 (*validation*)	603.78	593.42	315.62	110,629.46	99,617.3	0.03	1.11

indicate that primary production in a coastal ecosystem with a N:P ratio under 29 could be N-limited, while a value over 29 would be P-limited. As a result, values of PdC indicate that, on an annual average, primary production is limited by P concentrations, and this provides evidence of the importance of P to phytoplankton development. However, this condition become less evident in the summer period, where nitrogen may become the limiting factor (Sfriso et al., 1988).

Considering the seasonal variability, the trophic state follows the classic cycle of an aquatic ecosystem in a temperate climate (Facca and Sfriso, 2009; Guerzoni and Tagliapietra, 2006): in winter, primary production is low and the dynamics of nutrients, which are present in higher concentrations, are mainly influenced by loading and transport phenomena; in the spring time, solar radiation triggers the first phytoplankton blooms, which can be further stimulated or inhibited by the availability or lack of nutrients; nutrient concentrations show minimum values in the summer period, when phytoplanktonic blooms reach their peak and progressively decrease to minimum levels of abundance in autumn. In regards to the fractions of the main phytoplankton groups in the lagoon of Venice and PdC, diatoms represent the dominant group (72%), followed by flagellates (27%) and dinoflagellates (1%) (Guerzoni and Tagliapietra, 2006).

The trophic state of the Venice lagoon and PdC, analyzed within the context of the Water Framework Directive, has been evaluated through the abundance and composition of phytoplankton. Results from the study of ARPAV (2013b), based on the Multimetric Phytoplankton Index (MPI), indicate a 'Moderate' status of PdC. Results from another study (Facca and Sfriso, 2009), based on the Ecological Quality Ratio (EQR; Van De Bund and Solimini, 2007) calculated from values of diatom abundance and biomass, indicates a 'Poor' to 'Moderate' status of PdC. Finally, Critto et al. (2013) assessed the chemical state of the Venice lagoon by comparing measured total dissolved nitrogen (TDN) and total dissolved phosphorus (TDP) with the local regulatory Environmental Quality Standard (EQS) (DM, 1998), and for PdC the TDN exceeded the related EQR (200 μg/l).

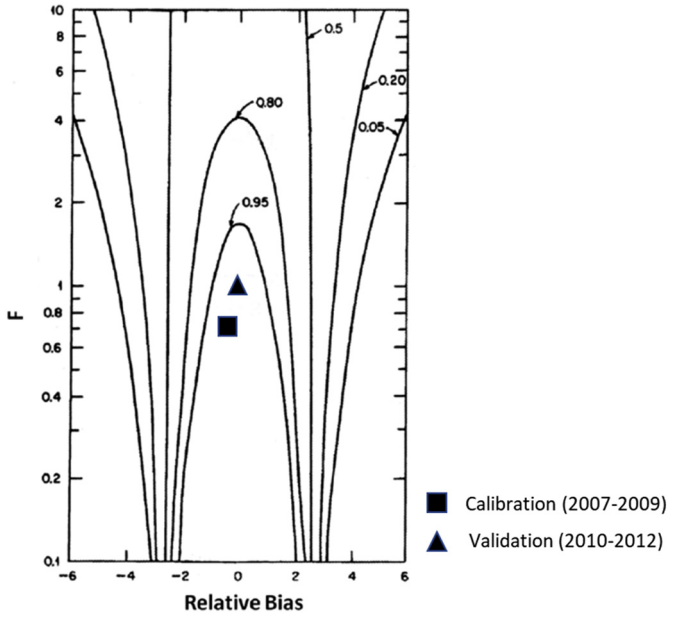


Fig. 6. Relative bias (rB) and F-test to compare means and variances of observed and modeled values of phosphorus. Isopleths indicate the probability that the predicted and observed distributions are similar, assuming normality.

2.2. Integrated modelling approach

The proposed integrated modelling approach allows to study and combine the three components identified in Fig. 1. As shown in Fig. 3 and summarized in Table 1, it adopts: (1) climate simulations generated by coupling of a General Circulation Model (GCM) with a Regional Climate Model (RCM) and by applying the tool CLIME for the downscaling of climate projections; (2) the hydrological model Soil and Water Assessment Tool (SWAT) for the modelling of river basin discharge and nutrient loads; and (3) the ecological model AQUATOX for the modelling of coastal phytoplankton. In this section, the environmental models and tools, and their parameterization, are described in detail.

2.2.1. Climate projections

In this study, climate projections produced by General Circulation Model/Regional Climate Model (GCM/RCM) nested simulations provide daily time series of maximum and minimum temperature and precipitation. A vast number of climate projections obtained from GCM/RCM nested simulations is available to researchers for impact assessment studies (Warszawski et al., 2014; Wilcke and Bärring, 2016) Given the small extent of the study area and in order to correctly represent the spatial variability of the climate conditions, GCM/RCM nested simulations available with the highest spatial resolution were selected, i.e. those generated by the coupling of the GCM CMCC-CM (Scoccimarro et al., 2011), with a spatial resolution of 0.75° × 0.75°, with the RCM COSMO-CLM (Rockel et al., 2008) under the configuration adapted to the Italian territory (Bucchignani et al., 2016; Cattaneo et al., 2012) with a spatial resolution of 0.0715° × 0.0715° (≈8 km × 8 km).

Along the past period, the CMCC-CM simulations under the "historical" experiment of the Climate Model Intercomparison Project 5 (CMIP5; https://cmip.llnl.gov/cmip5/), based on observed atmospheric composition changes (reflecting both anthropogenic and natural sources) and, for the first time, including reconstructed evolution of land cover (Hurtt et al., 2011) were considered in forcing COSMO-CLM. While the 'historical' runs cover much of the industrial period (from 1850 to 2005), Representative Concentration Pathways (RCPs; van Vuuren et al., 2011) were then assumed, since the year 2006, to simulate with CMCC-CM the future time horizons under alternative radiative forcing (Taylor et al., 2012), proxy of emission scenarios. The RCP4.5 (Thomson et al., 2011) is a stabilization scenario where total radiative forcing is stabilized, shortly after 2100, to 4.5 W m^{-2} (approximately 650 ppm CO_2-equivalent) by employing technologies and strategies to reduce GHG emissions. The RCP8.5 (Riahi et al., 2011) is a business as usual scenario and is characterised by increasing GHG emissions and high GHG concentration levels; it represents a rising radiating forcing pathway leading to 8.5 W m^{-2} in 2100 (approximately 1370 ppm CO_2-equivalent).

Simulation along 30-year time horizons, minimum period length for climatological studies as suggested by the World Meteorological Organization (WTO, 2007), were here used to compare mid-term (2041–2070) and long-term (2071–2100) future time frames against the control period 1983–2012. For the control period in this study,

Table 6
Values of relative bias (rB) and F for the considered parameters.

Parameter	Mean AQUATOX	Mean observations	St. dev. observations	Variance AQUATOX	Variance observations	rB	F
Sol. rad (Ly/d)	327.00	334.65	211.06	40,325.78	44,545.23	−0.03	0.91
Wind (m/s)	1.38	1.69	0.83	1.67	0.78	−0.38	2.14
DO (mg/l)	8.56	8.16	2.5	2.58	6.25	0.16	0.17
DIN (mg/l)	0.96	0.86	0.69	0.20	0.48	0.14	0.18
DIP (mg/l)	0.03	0.02	0.013	0.00011	0.00017	0.76	0.44
DIN:DIP	30.07	45.35	22.3	303.37	528.9	−0.66	0.34
Chl-a (μg/l)	3.44	3.5	5.58	35.7	31.09	−0.01	1.32

GCM/RCM simulations under the 'historical' experiment from 1983 to 2005 and then under one of RCP experiment from 2006 to 2012, according to the respective RCP considered in the future time frames for comparison, were considered.

Table 2 provides a summary of the scenarios adopted in this study.

RCMs are often used to dynamically downscale GCMs. However, in most cases, biases that prevent an appropriate reproduction of the observed climate conditions persist (Muerth et al., 2013). Thus, the application of a bias correction method is often recommended to feed impact models (Teutschbein and Seibert, 2012). In this study, the linear scaling bias correction method (Lenderink et al., 2007) was selected. This method consists in an adjustment of the mean value by adding a temporally constant offset to the simulated data (e.g. for temperatures), or by multiplying them by a temporally constant correction factor (e.g. for precipitation), usually at monthly level. This additive or multiplicative constant (delta) represents the average deviation between the monthly climatology of the simulated and the observed time series over the control period (Hempel et al., 2013). The bias correction was performed through the adoption of the tool CLIME, a GIS tool developed for climate data analysis and treatment. CLIME compared the observed values of temperature and precipitation available at each weather station (Castelfranco, Mogliano Veneto, Zero Branco; Fig. 2) with simulated values in the nearest point from the COSMO-CLM grid along the control period. The delta factors obtained for each calendar month over the control period were then applied to the daily series of the future periods, providing bias-corrected values of minimum and maximum air temperature and precipitation to be used as direct input for SWAT and to serve AQUATOX application (see next paragraphs).

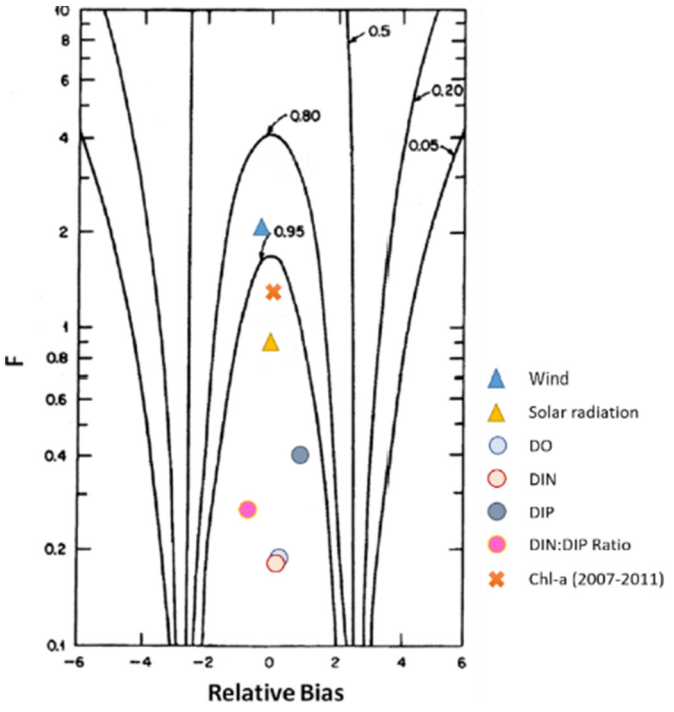


Fig. 7. Relative bias (rB) and F-test to compare means and variances of observed and modeled values of Wind, Solar radiation, DO, DIN, DIP, DIN:DIP and Chl-a. Isopleths indicate the probability that the predicted and observed distributions are the same, assuming normality.

Other climate variables such as relative humidity, solar radiation and wind speed influence surface hydrological processes and phytoplankton dynamics. While the simulation of these variables carries large uncertainties and generates very large discrepancies in magnitude and spatial patterns among different GCMs (Rockel and Woth, 2007; Seinfeld et al., 2016) and RCMs (Ahmed et al., 2013; Argüeso et al., 2013; Jakob Themeßl et al., 2011), there is limited amount of spatially explicit data over wide extents and with high spatial density to support model evaluation and thus the development and testing of bias-correction suited to the above variables. This is why bias-correction currently focuses mainly on precipitation and temperature (Bordoy and Burlando, 2013; Piani et al., 2010), which are found generally more impacting hydrological model results when comparing application using bias-corrected vs. non bias-corrected inputs (Haddeland et al., 2012; Papadimitriou et al., 2017). In this context, reconstructing the other variables by weather generator tools available within many impact (especially hydrological) models facilitates maintaining the temporal consistency among meteorological fields and limits the uncertainty from radiation, wind speed and relative humidity for which bias-correction procedures are less developed and rarely applied (Wilcke et al., 2013). Finally, expecting that the majority of changes in phytoplankton dynamics and composition will be associated to changes in temperature stratification and nutrient concentrations, which highly depend on temperature and precipitation (Winder and Sommer, 2012), it was decided to pay attention to modifications in these variables for the present study, and in order to subsequently feed the hydrological model SWAT and the ecological model AQUATOX.

2.2.2. Coastal watershed hydrology — SWAT model

The water cycle component was simulated with the semi-distributed hydrological model SWAT (Arnold et al., 1998). SWAT is a process-based eco-hydrological model developed to investigate the impacts of land management practices and climate on water, sediment and agricultural chemical yields in catchments over long periods of time. SWAT requires spatio-temporal explicit information about weather, soil properties, topography, vegetation, and land management practices. The hydrologic response units (HRUs) are the smallest computational units in SWAT, with unique land use, soil type and slope, and provide an efficient way of representing the spatial heterogeneity of a watershed (Kalcic et al., 2015). SWAT is widely used in climate change impact assessment studies (Cousino et al., 2015; Kim et al., 2016; Sellami et al., 2016), and it was here selected to simulate the climate-related pressures originating at the watershed level for the following reasons: (i) its suitability for representing hydrology and nutrient cycling processes over long periods of time (i.e. 100 years) (Borah et al., 2007; Marek et al., 2017); (ii) its capacity to be coupled with other environmental models (e.g. hydraulic, water quality, crop growth and ecological models) (Johnston et al., 2017); (iii) its ability to extensively represent land management practices and reproduce their effects (Epelde et al., 2015; White et al., 2016); (iv) the possibility to apply the model even to ungauged watersheds or with a low availability of

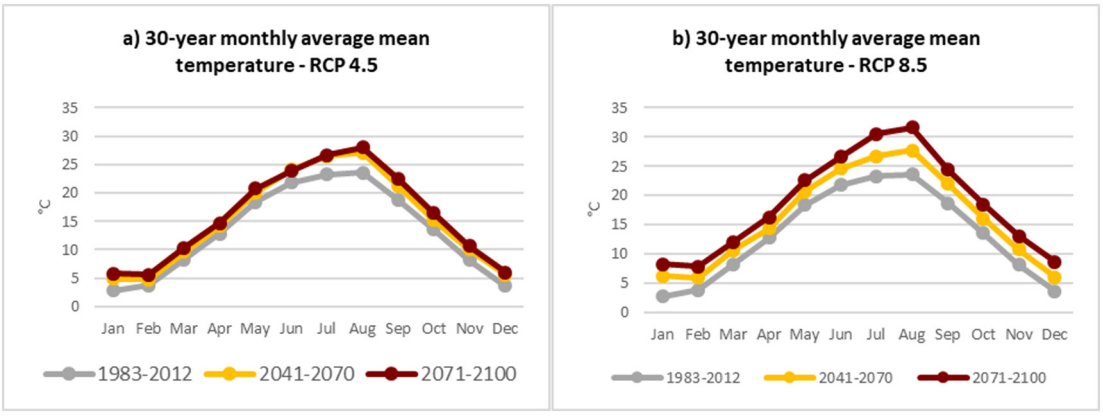


Fig. 8. 30-year monthly average of mean temperature in the control period (1983–2012) and the mid-term (2041–2070) and long-term (2071–2100) future projections for RCP 4.5 (a) and RCP 8.5 (b).

observed data (Cibin et al., 2014; Rafiei Emam et al., 2017; Sisay et al., 2017); and (v) strong technical support available from the SWAT community.

2.2.3. Coastal ecosystem — AQUATOX model

A coastal ecosystem model application was conducted by using the ecological model AQUATOX, which is a general aquatic ecological risk assessment model used to predict responses of aquatic life to multiple stressors such as water temperature, nutrients, sediments, and organic chemicals. AQUATOX is a mechanistic model that reproduces physical, chemical and biological processes at a daily or monthly time step of the simulation period within a volume of water. It can be applied to different aquatic environments such as rivers, lakes, ponds, reservoirs and estuaries. The model was selected to assess the effects of water temperature and nutrients loadings on the productivity and community structure of phytoplankton. Phytoplankton biomass in AQUATOX is modeled as a function of nutrient loadings, water retention time, photosynthesis, respiration, excretion, mortality, predation, sinking and sloughing. AQUATOX has been largely applied to simulate nutrient-related phytoplankton dynamics (Carleton et al., 2009) and climate change impacts on aquatic ecosystems (Taner et al., 2011) and it was selected here for its ability to: (i) assess impacts on the ecosystem from multiple stressors (i.e. water temperature, water discharge, nutrient loadings) for long time periods; (ii) select different species of phytoplankton (i.e. diatoms, dinoflagellates, and cyanobacteria); and (iii) easily utilize the outputs of SWAT (i.e. water discharge, nutrient loadings).

2.3. Model parameterization

2.3.1. SWAT model parameterization for modelling the ZRB

The hydrology and water quality conditions of the Zero river were modeled with SWAT using the following data for the period 2007–2012: daily time series of meteorological data (i.e. precipitation, maximum and minimum temperature, wind, solar radiation, relative humidity) from the 3 weather stations (i.e. Castelfranco Veneto, Zero Branco, Mogliano; Fig. 2) well representing the climate across the basin; a digital elevation model at 5 m × 5 m spatial resolution (Regione Veneto, 2013); a land-use map at 100 m × 100 m spatial resolution (ARPAV, 2009b); a soil map with soil geomorphological and textural characteristics at 500 m × 500 m spatial resolution (ARPAV, 2001); and agricultural management practices for the selected cultivations (corn, soy, winter wheat) (Bonetto and Furlan, 2012; Regione Veneto, 2014; Tassinari, 1976). In addition, for model calibration and validation purposes, we used: daily time series of streamflow (MAV, 2013); daily time series of inorganic nitrogen concentrations (i.e. nitrate and ammonia) (ARPAV, 2013c); and seasonal values (4 values for each year) of inorganic phosphorus concentrations (ARPAV, 2013c).

Due to the lack of continuous data, the influence of groundwater recharge from the bordering watersheds was modeled with an additional constant contribution. For the same reason, the influence of point-source pollution (Waste Water Treatment Plant, WWTP; Industrial discharges) on nutrient loads was simulated with an additional constant contribution. Values were obtained from previous studies (Salvetti et al., 2008; ARPAV, 2007, 2009a; Regione Veneto, 2013, 2014). A

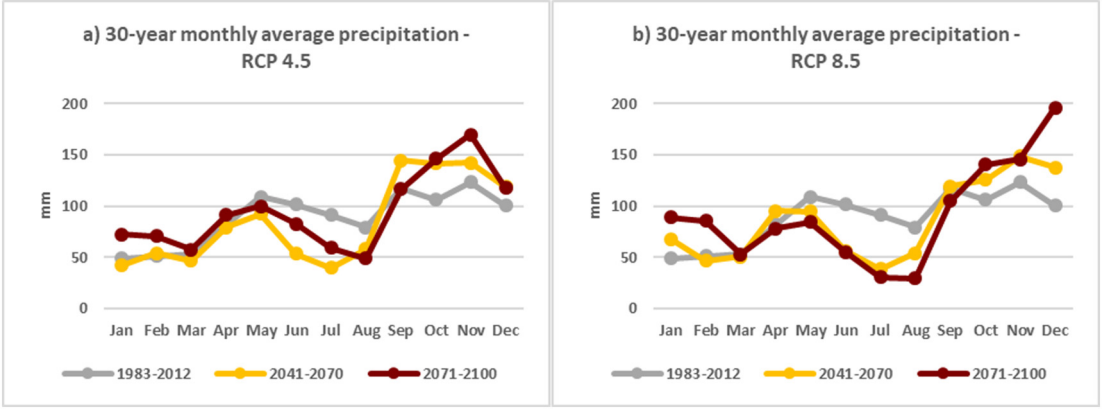


Fig. 9. 30-year monthly average of mean precipitation in the control period (1983–2012) and the mid-term (2041–2070) and long-term (2071–2100) future projections for the RCP 4.5 (a) and RCP 8.5 (b).

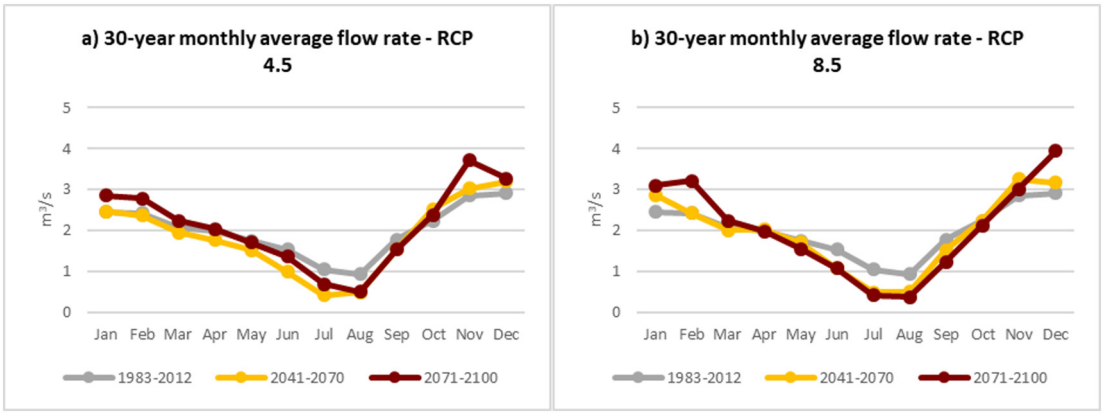


Fig. 10. Flow rate differences in the 30-year monthly average of the control period (1983–2012) and the mid-term (2041–2070) and long-term (2071–2100) future projections for the RCP 4.5 (a) and RCP 8.5 (b).

detailed description of model parameterization is provided in the supplementary material A-I.

The SWAT model of the ZRB was run for the period 2007–2012. A warm-up period of three years (2004–2006) preceded the simulation. The length of the warm-up period was selected to allow the model of SWAT to reach a steady-state, as suggested in the SWAT manual (Arnold et al., 2012). The calibration period was set from 2007 to 2009 and the validation period from 2010 to 2012. The SWAT model outputs relative to water discharge (Q, $m^3\ s^{-1}$), Nitrogen from nitrate (N-NO_3^-, tons) and ammonium (N-NH_4^+, tons), and phosphorus from orthophosphate (P-PO_4^{3-}, tons) loads were provided as monthly time series and were used as input for the AQUATOX model.

The calibration based on water discharge and inorganic nitrogen (N-NO_3^-, N-NH_4^+) loads was performed with the software SWAT-CUP (Abbaspour, 2014). The data regarding phosphate (PO_4^{3-}) concentrations were not sufficient to perform a meaningful monthly calibration. As a result, the overlap between observed and modeled data distribution attributes, based on relative bias (rB) and variance ratio (or F-test), was conducted. The parameters adjusted for calibration are listed in Table 3. A detailed description of the methods for the evaluation of the model performance is provided in the supplementary material A-II.

2.3.2. AQUATOX model parameterization for modelling the PdC

AQUATOX was used to represent phytoplankton dynamics and composition in PdC. Given the purpose of the study to investigate the impacts of climate change along the land-water continuum, and the impossibility for AQUATOX to run when a waterbody goes dry (i.e. water volume = 0 m^3), the effect of tide and sea level rise has been neglected. As a result, PdC has been simulated as an enclosed waterbody with a constant volume of water influenced by the freshwater discharge and nutrient loading coming from the ZRB. The model was run for the period 2007–2011, as data necessary for the evaluation of the model performance were not available for the year 2012. Two years of warm-up period (2005–2006) were selected, as one year was not sufficient to bring the system to a steady-state. For the modelling of PdC in AQUATOX, the following parameterization data were used to initialize the model: site length (L) and width (W) were set at 4.5 km and 0.8 km, respectively; surface area was set at 3.6 km^2; mean and maximum depth were set respectively at 1 m (H) and 3.2 m (H_{max}). Accordingly, the volume (V) of the system was set constant at $3.6 \times 10^6\ m^3$ ($V = L \times W \times H$). Evaporation was set constant to the value of 0 mm/year as it was assumed in balance with the direct precipitation over the lagoon and therefore the two variables tend to cancel each other out (Zirino et al. 2014). Monthly means of water discharge ($m^3\ s^{-1}$) and inorganic forms of nitrogen and phosphorus loads (kg $month^{-1}$) were obtained from the SWAT simulation of the ZRB; monthly values of water temperature were obtained from the monitoring network SAMANET (Ferrari et al., 2004); the annual average and maximum-minimum range of wind speed (m/s) and light intensity at the surface (Ly day^{-1}) were determined through meteorological data obtained from ARPAV (2013a) and used as input for the AQUATOX weather generator to generate daily values of the two variables; latitude, necessary for the calculation of daily values of light intensity, was set at 45°3′N; time-average constant values were assumed for pH (7.9); initial concentrations of nutrient concentrations were obtained from MELa3 monitoring campaign (MAV and CVN, 2002) and set to average

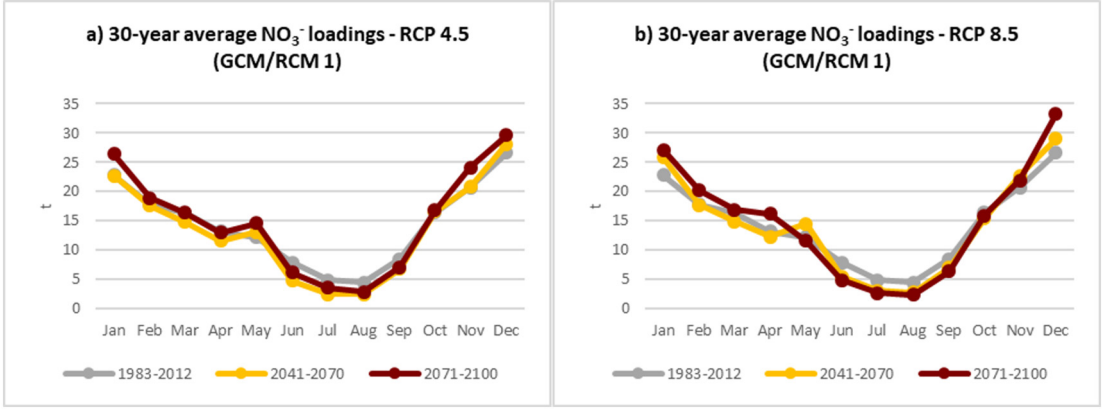


Fig. 11. Nitrate loadings differences in the 30-year monthly average of the control period (1983–2012), mid-term (2041–2070) and long-term (2071–2100) future projections for scenarios RCP 4.5 (a) and RCP 8.5 (b).

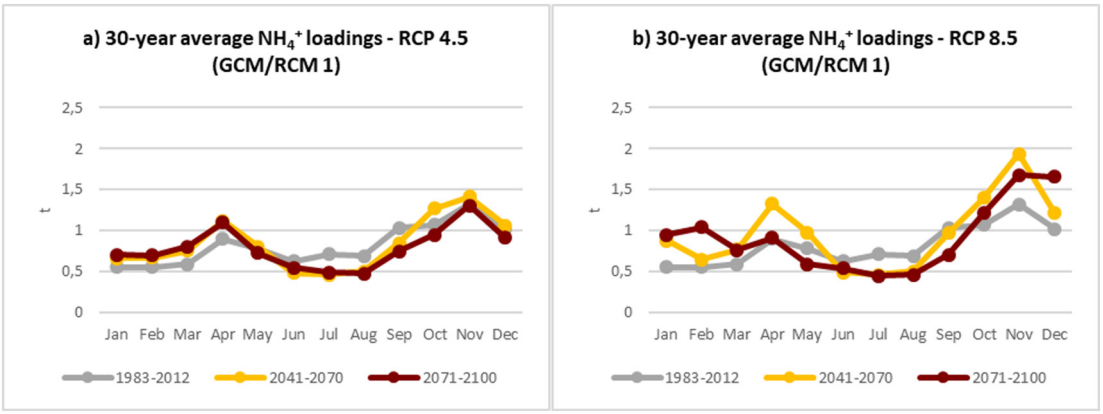


Fig. 12. Ammonium loading differences in the 30-year monthly average of the control period (1983–2012), mid-term (2041–2070) and long-term (2071–2100) future projections for scenarios RCP 4.5 (a) and RCP 8.5 (b).

values of winter 2007: 0.16 mg/l for $N\text{-}NH_4^+$, 0.90 mg/l for $N\text{-}NO_3^-$, and 0.03 mg/l for $P\text{-}PO_4^{3-}$. Initial concentration of dissolved oxygen (DO, mg l^{-1}) and salinity (PSU) were obtained from SAMANET monitoring network (Ferrari et al., 2004) and set to 10.45 mg/l and 15 ppt; total suspended solids (TSS, mg l^{-1}) were set equal to 8 mg l^{-1}, based on average values of calm conditions (without the effect of wind and waves) in the lagoon of Venice (Thetis, 2006).

The modeled ecosystem is composed of detritus, phytoplankton, and zooplankton. Detritus is subdivided into suspended particulate refractory detritus, suspended particulate labile detritus, sediment refractory detritus, sediment labile detritus and dissolved detritus. As no site-specific information was available for detritus in the sediment, both refractory and labile initial values were set to 0, leaving to AQUATOX the task of bringing them to a steady-state, as suggested in the AQUATOX manual (Park and Clough, 2009). Initial values of detritus in the water column were obtained from values of total organic carbon (TOC) available from the MELa3 monitoring campaign, selecting the average of the measurements of January for the years 2007–2009 (3.0 mg/l) as initial condition. Nine phytoplankton compartments, 8 diatoms (D) and 1 cyanobacteria (CB), were added to represent the possible evolutions of phytoplankton biomass and composition in present and future conditions (Table 4). In this study, the default organisms in the AQUATOX library (release 3.1) having the most similar characteristics to the species of the lagoon of Venice (Facca, Sfriso, and Ghetti 2004) were selected. *Navicula* and *Cyclotella* spp. are species commonly found in the waters of the lagoon of Venice. *Fragilaria* sp. was selected as a species that can be found in transitional water ecosystems. A species of Cyanobacteria was implemented to observe if future environmental conditions would favor blooms of cyanobacteria. The species *Microcystis* sp. was selected from the AQUATOX database as a species that is fairly salt-tolerant and able to produce microcystin toxins that can be stable and persistent in transitional ecosystems (Gibble and Kudela, 2014). A more detailed description of phytoplankton species is provided in the supplementary material A-III.

Concerning the zooplankton, the species *Acartia clausi* was selected as representative in PdC: initial concentrations were set to 0 mg/l, and the warm-up period was used to bring the species to equilibrium. Moreover, AQUATOX uses "seeds" of zooplankton, very small concentrations (1E−05 mg/l dry) that are added into the system on a daily basis for preventing the extinction of the species. Model performance was evaluated for the period 2007–2011 still by comparing observed and modeled data distribution attributes based on relative bias (rB) and variance ratio (F-test). A detailed description of the methods for the evaluation of the model performance is provided in the supplementary material A-II. Specifically, the model performance has been evaluated on the following parameters: solar radiation and wind speed to evaluate the performance of the weather generator built in AQUATOX; dissolved oxygen (DO); dissolved inorganic nitrogen (DIN); dissolved inorganic phosphorus (DIP); DIN:DIP ratio; Chlorophyll-a (Chl-a). DIN and DIP were evaluated for the period 2007–2009 as further data were not available.

2.3.3. Modelling assumption and caveats in future scenarios

To simulate future physical, chemical and ecological conditions of the study area, a number of assumptions and caveats were taken. First, some meteorological variables (i.e. wind speed, solar radiation, relative humidity) were kept constant in hydrological modelling. Statistical parameters of monitored data for these variables along the calibration and validation periods (2007–2012) were used in the SWAT Weather Generator to produce daily data consistently with temperature and

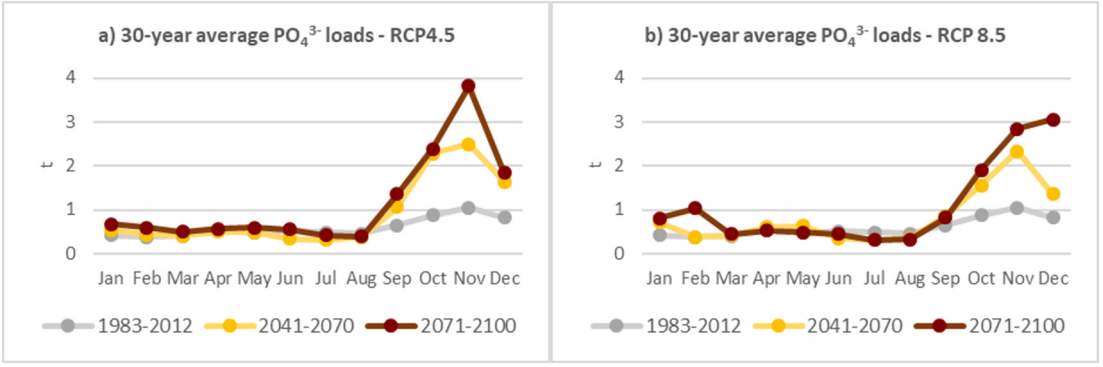


Fig. 13. Inorganic phosphorus loading differences in the 30-year monthly average of the control period (1983–2012), mid-term (2041–2070) and long-term (2071–2100) future projections for scenarios RCP 4.5 (a) and RCP 8.5 (b).

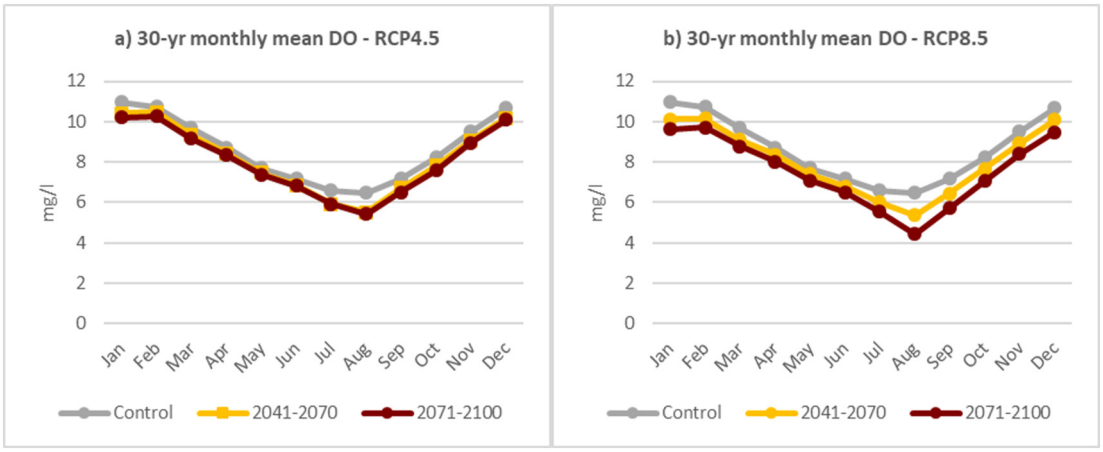


Fig. 14. Differences in the 30-year DO monthly mean between the control period (1983–2012), and the mid-term (2041–2070) and long-term (2071–2100) future projections for RCP4.5 (a) and RCP8.5 (b).

precipitation fields given as input to the model. In AQUATOX, mean and range of wind speed and solar radiation computed by the AQUATOX weather generator were kept constant for both mid-term and long-term periods. Second, land-use, agricultural practices and other anthropogenic emissions (i.e. WWTP, industrial discharges) in the Zero river basin remained unchanged. Third, the effects of sea level rise and human infrastructures such as MOSE project at the inlets of the lagoon of Venice have been neglected. Finally, given the absence of high-resolution water temperature projections for the lagoon of Venice for RCP4.5 and RCP8.5, projected water temperature was computed by using a linear regression between monitored air temperature and water temperature ($R^2 = 0.99$).

3. Results and discussion

3.1. Calibration of the SWAT model

Calibration under a monthly time step for the 2007–2009 period (Fig. 4) produced satisfactory results for water discharge (NSE = 0.58, $R^2 = 0.63$), nitrate (NSE = 0.60, $R^2 = 0.80$) and ammonium (NSE = 0.51, $R^2 = 0.59$), based on the performance ratings developed by Moriasi et al. (2007).

Validation was performed for the 2010–2012 period (Fig. 5) and resulted in lower NSE for flow rate (NSE = 0.20, $R^2 = 0.61$) and nitrate (NSE = 0.25, $R^2 = 0.65$), related to an extreme precipitation event in October–November 2010 and an underestimation of flow rate during the 2011–2012 autumn-winter, characterised by very low precipitations (Fig. 5a, b). The low performance (NSE = −0.10, $R^2 = 0.25$) of ammonium during validation period (Fig. 5c) are also attributed to underestimated flow rate during the 2011–2012 autumn-winter period.

Regarding the calibration and validation for phosphorus loads, the computed relative bias (rB) and variance ratio (F-test, F) indicated in Table 5 and Fig. 6 show that observed and modeled distributions are similar, assuming normality. Moreover, the performance of AQUATOX for DIP concentration of PdC (discussed in the next section) supports the acceptability of these results for the purposes of this study.

3.2. Performance evaluation of the AQUATOX model

Wind speed, solar radiation, DO, DIN and DIP concentrations, and Chl-a concentrations were used to evaluate the performance of the AQUATOX model of PdC. Relative bias (rB) versus variance ratio (F-test, F) were used to evaluate the performance of the model. Table 6 and Fig. 7 summarizes the outcomes of the overlap test through the computed values of rB and F.

In general, it is possible to observe that the majority of modeled variables are in good agreement with the observed data. Discrepancies between modeled and observed values are detected for dissolved inorganic phosphorus and DIN:DIP ratio. The model slightly overestimates phosphorus concentrations over the year. This justifies the higher value of rB (0.76) in the overlap test. A plausible explanation may be the difficulty of AQUATOX in modelling the complex dynamics of nutrient

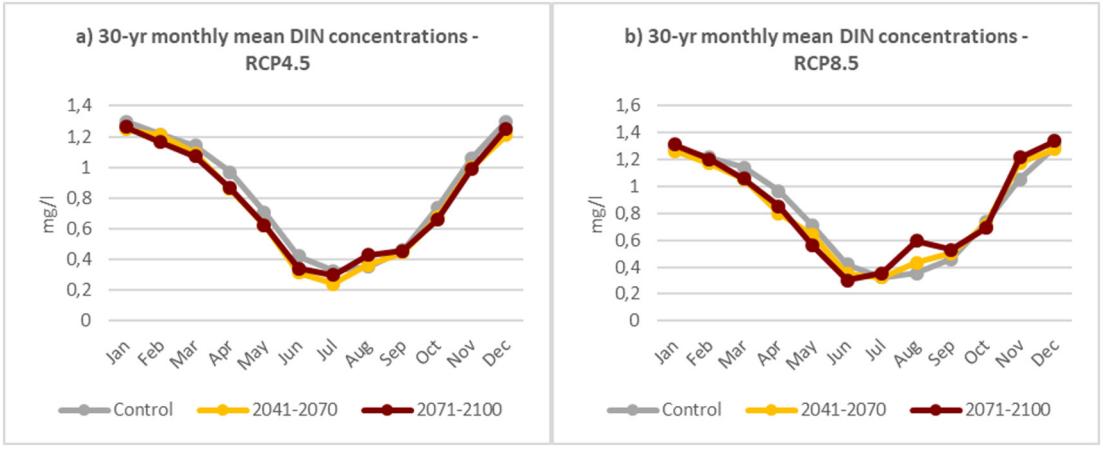


Fig. 15. Differences in the 30-year DIN monthly mean between the control period (1983–2012), and the mid-term (2041–2070) and long-term (2071–2100) future projections for RCP4.5 (a) and RCP8.5 (b).

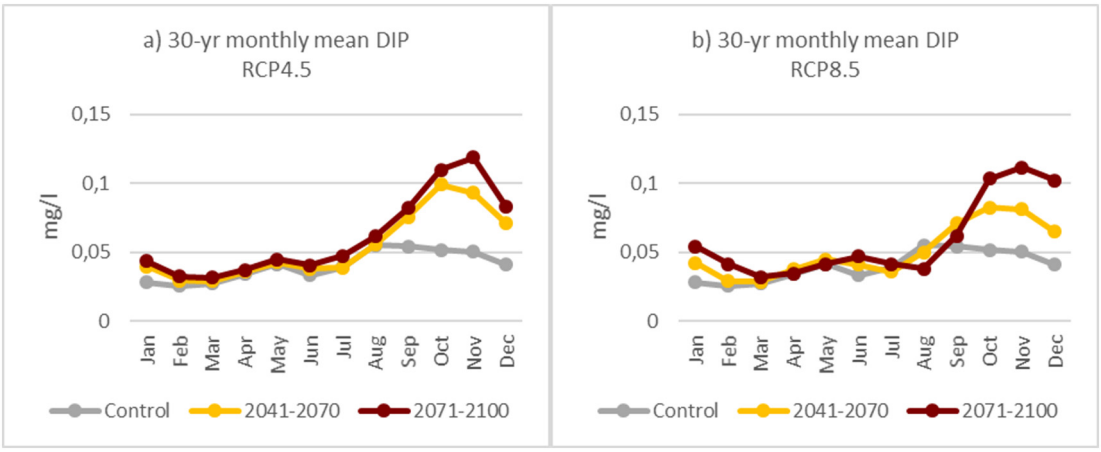


Fig. 16. Differences in the 30-year DIP monthly average concentrations between the control period (1983–2012), and mid-term and long-term projections for RCP4.5 (a) and RCP8.5 (b).

between sediment and water column, specifically the removal of phosphate, as explained in Zirino et al. (2016b). Consequently, the modeled DIN:DIP ratio is slightly underestimated, as demonstrated by the rB value (−0.66). However, the seasonal fluctuations are preserved.

3.3. Watershed responses to climate change

The projected modifications in temperature and precipitation due to climate change, as shown in Figs. 8 and 9 and average over the ZRB, are expected to produce variations in freshwater discharge and nutrient loads both in the mid-term (2041–2070) and long-term (2071–2100), as predicted also in other studies (Alam and Dutta, 2013; Amin et al., 2017; Bosch et al., 2014; Tong, 2007; Whitehead et al., 2009). In this section, mid- and long-term projections of water flow and nutrient loadings for RCP4.5 and RCP8.5 are discusses in relation to the control period (1983–2012). Water discharge projections do not show any change in the annual average, with 30-year yearly mean stable at 2 $m^3 s^{-1}$. However, seasonal changes are detected. SWAT projects an increase in the late autumn-winter flow of 5% for RCP4.5 and 8% for RCP8.5 in the mid-term, and of 8% for RCP4.5 and 20% for RCP8.5 in the long-term. In summer, the reduction of flow is above 40% for RCP4.5 and RCP8.5 in both mid-term and long-term (Fig. 10). Results are consistent with precipitation projections, which indicate an increase in winter and a reduction in summer, coupled with an increase in summer evapotranspiration due to the higher temperatures. Results agree with other studies suggesting that future water flow will be shaped by an increase in winter and a decrease in summertime (Bastola, Murphy, and Sweeney 2011; Cervi et al., 2018; Middelkoop et al., 2001; Radchenko et al., 2017).

As assessed by other authors, nutrient loadings are also affected by changes in climate and consequent modifications in hydrological regimes (Chang et al., 2001; Jeppesen et al. 2009a,b). In this study, projections of N-NO_3^- loads are consistent with changes in precipitation ($R >$ 0.9) for both RCP4.5 and RCP 8.5, matching with other studies (Martínková et al., 2011; Shrestha et al., 2012). Results indicate an increase in the average yearly loadings over the 21st century, with values that rise up to 5% for both RCP4.5 and RCP8.5 by the end of the century (Fig. 11). Seasonal changes are more evident, especially in the winter and summer months. Mid-term projections for both RCP4.5 and RCP8.5 show a slight increase below 10% in winter loadings for both RCP4.5 and RCP8.5, and a decrease of 35–40% in summer. Long-term projections for RCP4.5 show only slight changes in relation to mid-term projections, except winter months that experience a further increase of 9%. Differently, long-term projections for RCP8.5 indicate greater changes, with a 22% increase in winter and a 44% decrease in summer in relation to the control period.

The SWAT application for ZRB indicates that ammonium loads tend to increase in the winter-spring period and reduce in summer in case of RCP4.5. Under the RCP8.5, ammonium shows an increase from October to April, and a decrease in the summer period. These results indicate that ammonium loadings follow the projected changes in precipitation, and especially intense precipitation can facilitate erosion and runoff of soil particles and the absorbed ammonium, as discussed by other authors (He et al., 2014). It can be also noted that, for both RCP4.5 and RCP8.5, higher loads of NH_4^+ were observed in the mid-term period

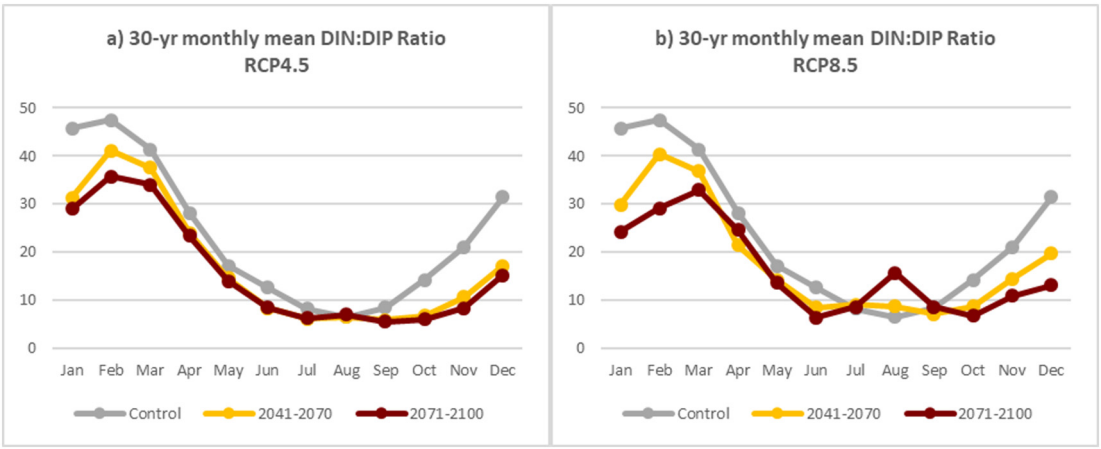


Fig. 17. Differences in the 30-year DIN:DIP monthly mean between the control period (1983–2012), and the mid-term and long-term projections for RCP4.5 (a) and RCP8.5 (b).

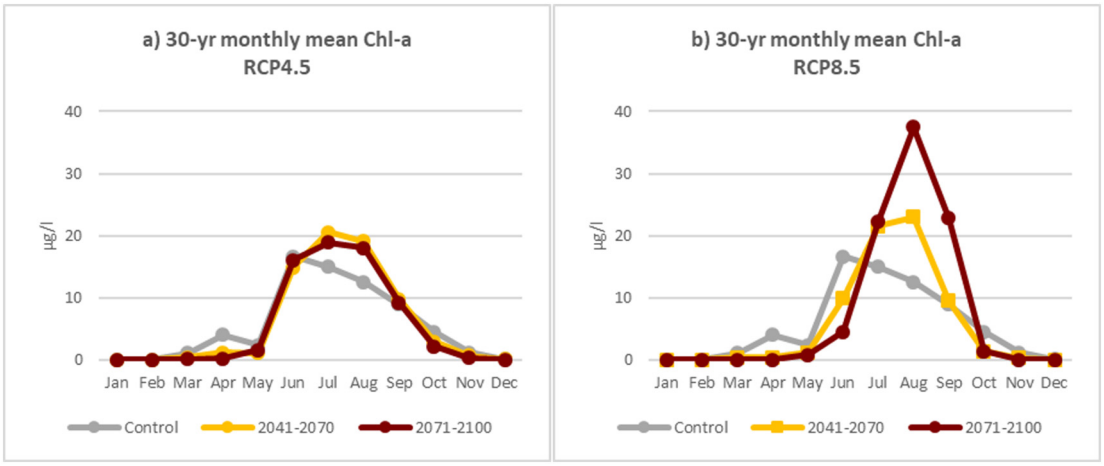


Fig. 18. Differences in the 30-year monthly average Chl-a concentrations between the control period (1983–2012), and mid-term (a) and long-term (b) projections (2071–2100) of RCP4.5 and RCP8.5.

(Fig. 12). This shows the relevant role of temperature and soil water content to nitrogen dynamics, according to the physical and chemical processes implemented in the SWAT model, such as mineralization, nitrification, and volatilization, which are influenced by temperature and available water, and reach their optimal values within a range of temperature and humidity in the soil (Aranibar et al., 2004; Hedin et al., 1998).

Finally, changes in phosphorus loadings were also assessed (Fig. 13). Results indicate changes in the magnitude of the autumn-winter loads of 50–60% in the mid-term, and 90% to 300% in the long-term for, respectively RCP4.5 and RCP8.5. This is caused by intensified leaching and erosion processes caused by increased precipitation in the autumn-winter period. According to other studies (such as Shimizu et al. (2011), which studied the effects of climate change on nutrient discharge in a coastal area of Japan), SWAT results indicate that phosphorus loadings are highly correlated with precipitation ($R > 0.8$). Moreover, drier conditions in the summer might exacerbate the soil erosion in the autumn season, and consequently increase the runoff of sediments and adsorbed mineral forms of phosphorus. Finally, results indicates an enrichment of the topsoil in inorganic phosphorus, which can be caused by an excess of fertilization in agricultural practices and by accelerated mineralization and sorption to clay particles due to increased temperatures, as identified in Jennings et al. (2009) and Lapp et al. (1998).

3.4. Physical and chemical responses of the coastal ecosystem

In this section, future mid-term and long-term projections of water quality parameters (DO, DIN, DIP, DIN:DIP ratio) in the PdC are compared towards the control period (1983–2012).

Simulated results of dissolved oxygen (DO) indicate that concentrations feature a decrease in both mid- and long-term period for RCP4.5 and RCP8.5. DO registers a 36% decrease in summer, and a 5% to 15% decrease in winter (Fig. 14). According to others studies investigating coastal and oceanic waters (such as Schmidtko et al. (2017) and Rabalais et al. (2009)), this behavior suggests rise in temperature (due to climate change) as a fundamental factor of oxygen reduction in PdC, since the solubility of oxygen decreases while temperature increases. Moreover, given that AQUATOX does not capture the high excursions that take place in PdC between day and night, a similar decrease in 30-yr monthly averages might imply an increase in daily hypoxic conditions.

In relation to dissolved inorganic nitrogen (DIN), the main changes can be identified in long-term projections for both RCP4.5 and RCP8.5.

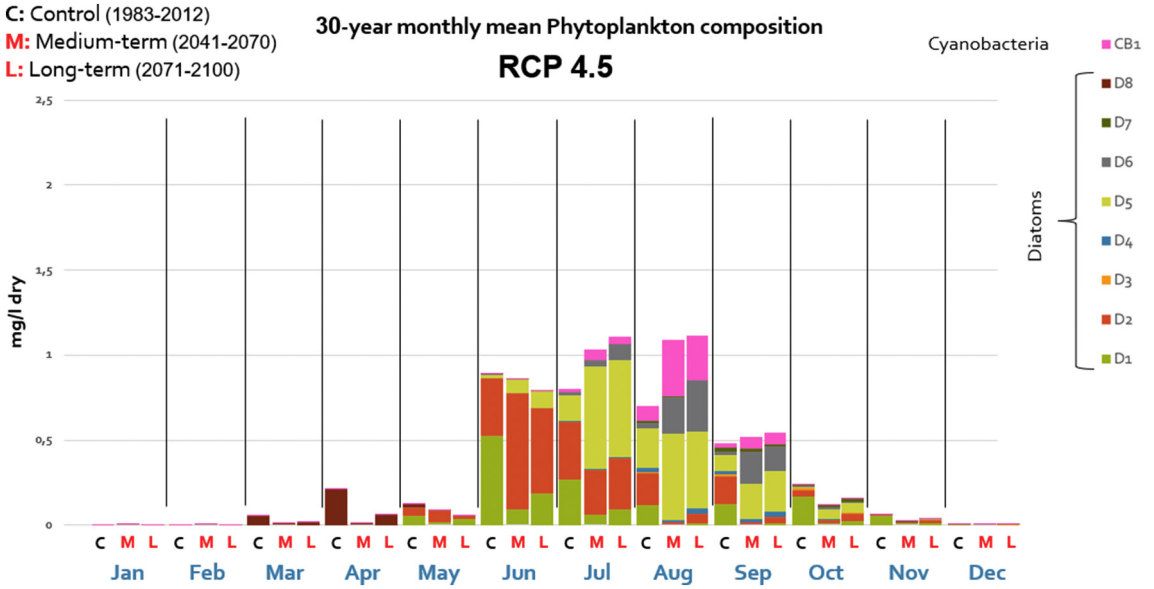


Fig. 19. Comparison of abundance of different species of phytoplankton between control period and mid- (2041–2070) and long-term (2071–2100) periods for RCP4.5.

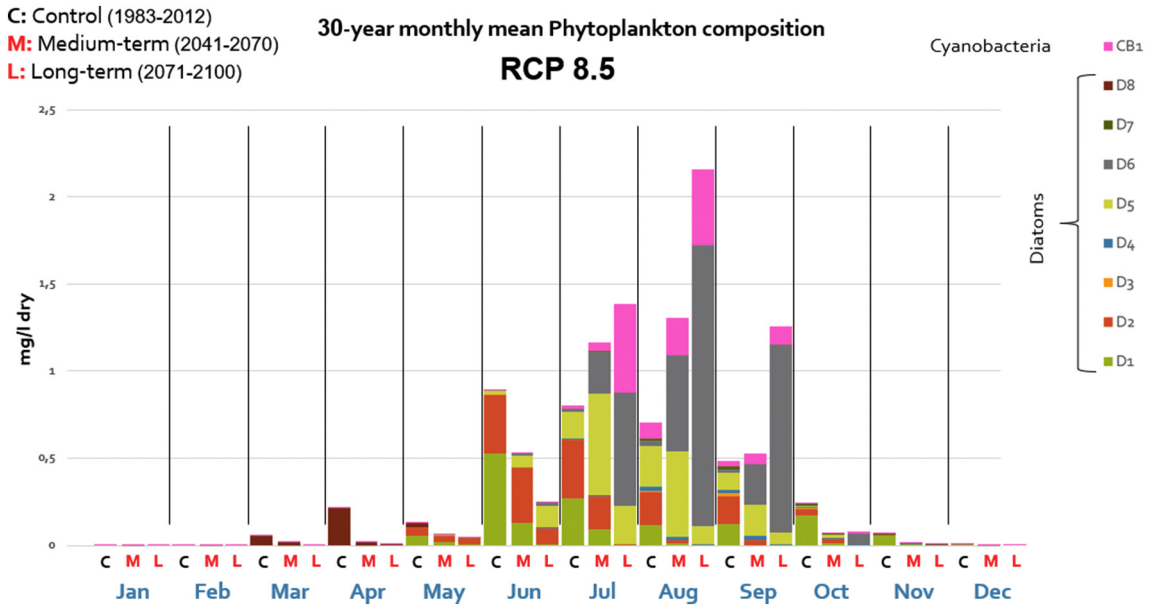


Fig. 20. Comparison of abundance of different species of phytoplankton between control period and mid- (2041–2070) and long-term (2071–2100) periods for RCP8.5.

AQUATOX results indicate an increase of DIN concentration in summer (Fig. 15), with changes of 23% for RCP4.5 and 30% for RCP8.5 in relation to the control period. This is caused by an increase of NH_4^+ concentrations, as higher temperatures increase the release of NH_4^+ from the sediment because of higher mineralization rates and a smaller aerobic zone (Gonenc and Wolflin, 2004). Indeed, ammonium is also controlled by changes in DO: when DO decreases, nitrification process decrease and, as a result, ammonium concentration increase (Elshemy, 2013). In the other seasons DIN features a general stability in its concentration levels, indicating that the surplus of DIN coming from the ZRB is assimilated by phytoplankton and higher trophic levels of the system (i.e. zooplankton).

Dissolved inorganic phosphorus (DIP) concentrations for both RCP4.5 and RCP8.5 reflect the changes in phosphorus loadings coming from the ZRB. It is possible to observe an increase of phosphorus concentrations in the autumn-winter period, with a mid-term increase of 50% for both RCP4.5 and RCP8.5, and a long-term increase of 70% for RCP4.5 and 109% for RCP8.5. Differently, summer concentrations remain similar to the levels of the control period (Fig. 16).

Future projections of DIN:DIP ratio estimated by AQUATOX reflect the increase in phosphorus concentration in autumn and winter (reducing the DIN:DIP ratio) and the increase in ammonium in august (Fig. 17). Given the overestimation of phosphorus in the model and the greatest changes in the DIN:DIP ratio happening in the winter season, results do not suggest relevant effects on the composition of phytoplankton in PdC. However, as suggested by other studies (Zirino et al., 2016b), the events in which nitrogen become the limiting nutrient may become more frequent than in present times. Moreover, it was not possible to estimate changes in silicates (Si), as both SWAT and AQUATOX do not present this feature. The presence of silicate, however, is fundamental for the growth of diatoms (Egge and Aksnes, 1992), and changes in the N:Si and P:Si nutrient ratios can modify the competitive advantage of phytoplankton species. Indeed, as N and P are discharged in excess over Si (due to anthropogenic activities) non-diatoms species (often undesirable ones) might develop strongly (Choudhury and Bhadury, 2015; Garnier et al., 2010).

3.5. Phytoplankton responses of the coastal ecosystem

According to the 30-year averages of Chl-a concentrations (Fig. 18), total phytoplankton biomasses increase in both investigated scenarios, particularly in the long-term period of RCP8.5. In the RCP4.5 scenario, the spring peak disappears as the representative species for that period (D8, Table 3) stopped growing. Moreover, AQUATOX results indicate an increase in summer for Chl-a concentrations of 17% and 13% for respectively mid- and long-term periods of RCP4.5, and an increase of 20% and 40% for respectively the mid-term and long-term under RCP8.5 scenario. Yearly average concentrations rise from 66.44 $\mu g\ l^{-1}$ (control period) to 67.7 $\mu g\ l^{-1}$ (2041–2070) and 89.48 $\mu g\ l^{-1}$ (2071–2100). It is also observable a shift in the peak of Chl-a, from June to July for RCP4.5, and to August for RCP8.5. It is important to consider the fact that Chl-a values are representative of the phytoplankton composition in the system and they cannot model adaptation or addition of new, more tolerant species.

Differences are also noticeable in the composition of phytoplankton (Figs. 19 and 20). In the control period, phytoplankton is mainly composed of D1, D2, D5, and D8 (Table 3), representative of the spring bloom in PdC. The major changes happen in the summer months, where the abundances of Cyanobacteria (CB1) and diatoms adapted to warmer temperatures (D6) increase noticeably. Another observation is that Navicula (D1), a common diatom in PdC and other areas of the lagoon of Venice, tends to disappear in every future scenario. These results illustrate how different climate conditions will promote the growth of species which are more resistant to warm temperatures, and inhibit the growth of the species that currently dominate the composition of phytoplankton. Substantial stability in the DIN:DIP ratio in the blooming period does not promote growth of phytoplankton with different nutrient ratios, such as D3, D6 and D7 (Table 3). Thus, results indicate that major changes in the phytoplankton community might be caused by higher temperatures, while changes in nutrient loadings might be a secondary driver of change.

4. Conclusions

This study presents an integrated modelling approach for assessing potential medium (2041–2070) and long-term (2071–2100) effects of climate change, represented by modifications in temperature and precipitation, on the productivity and community structure of coastal phytoplankton at a local scale. Results based on the modelling through SWAT indicate that the projected changes in climate (IPCC, 2013) will likely alter current hydrological conditions of the Zero river basin, and this can have implications for future nutrient loading patterns. SWAT

results indicate that water flow will likely increase in winter and decrease in summer, and so will do nutrient loads discharged into Palude di Cona. These findings are in agreement with other case studies that assessed the effect of climate change on the hydrology (Bosch et al., 2014; Whitehead et al., 2009) and nutrient dynamics (Jeppesen et al. 2009a,b; Martínková et al., 2011; Shimizu et al., 2011) of river basins. Moreover, according to other studies (Moss et al., 2011; Rabalais et al., 2009; Winder and Sommer, 2012), the results of AQUATOX suggests that the ecological impact of modified loads and warmer temperatures will likely increase the frequency of eutrophication events in summer, with higher peaks of phytoplanktonic biomass, and change the phytoplankton composition. Specifically, results indicate that species adapted to warmer waters will dominate and replace current species, and that will generate greater blooms. In addition, the increased frequency of nitrogen limited conditions, together with a greater availability of phosphorus, may favor the growth of nitrogen-fixing cyanobacteria, as pointed out also by other authors (Dolman et al., 2012).

The implemented integrated modelling approaches necessitated some assumptions and simplifications that added uncertainty to the study. First, statistical attributed of meteorological drivers such as wind, relative humidity and solar radiation were not changed from those of the observed values and, due to the uncertainty of projected values of these variables (Papadimitriou et al., 2017; Rockel and Woth, 2007; Seinfeld et al., 2016) and the lack of well assessed methods for their bias-corrections (Bordoy and Burlando, 2013; Piani et al., 2010; Terink et al., 2010), these were created by weather generator tools. As relative humidity, radiation and wind speed have an important role in the hydrological processes and on the hydrodynamics of the lagoon, as well as on the biological processes of phytoplankton, this can be considered a potential area of improvement. Second, only results from the nesting between one GCM and one RCM scenario are presented in this study. However, considering that the obtained results are highly dependent on the assumptions of the selected GCM and/or RCM, more than one GCM/RCM combination should be applied to the integrated methodology in order to consider the uncertainty due to climate change projections (Al-Safi and Sarukkalige, 2017), and this could be part of future research activities. Third, SWAT and AQUATOX do not allow to model silicates (Si). Therefore, it was not possible to assess changes in Si concentrations and the consequent effects on Si—N and Si—P ratios. Si is fundamental for diatoms (Egge and Aksnes, 1992), which represent the main phytoplanktonic group in PdC, and changes in concentrations could change the composition of phytoplankton communities (Garnier et al., 2010). This aspect should be further explored in following studies. Fourth, land-use, agricultural activities and nutrient loadings from other anthropogenic sources (e.g. WWTPs) were kept as constant. Although the study focused on the effect of climate change, in particular temperature and precipitation, the effect of land-use change and other non-climatic pressure may be important drivers and therefore the integration of projections on land-use and other human influences (e.g., population dynamics, agronomic practices) on the environment could add further value to the modelling approach (Holman et al., 2017). Moreover, global events such as the projected P shortage anticipated for the coming decades, and the potential reduction in fertilizers were not considered (Glibert et al., 2014). Finally, the effect of tide, sea level rise and hydraulic infrastructures (e.g. Mose Project) were neglected in this study but could have important consequence on the hydrodynamics, and therefore on the ecosystems of the lagoon in the future.

The study demonstrates the utility of integrating climate scenarios and environmental models to project the impacts of multiple stressors on abiotic and biotic components of aquatic ecosystems. Specifically, SWAT and AQUATOX offer valuable tools to project the impacts of climate change on watersheds and on aquatic ecosystems. Both models have been applied in assessing climate change independently; however, to our knowledge, they have not previously been coupled together for this purpose. In conclusion, to further improve this integrated modelling approach and use it for planning/management purposes, potential improvements include: (1) adoption of additional GCM/RCM scenarios to assess the uncertainty coming from climate projections and the sensitivity of ecological parameters to climate drivers; (2) incorporating additional atmospheric drivers (i.e. wind speed, solar radiation, relative humidity) after applying appropriate bias-correction to the simulated time series; (3) implementing a hydraulic model able to simulate the specific hydrodynamics of the lagoon, especially the interaction with the sea; (4), integrating land-use change scenarios and other models capable of simulating and projecting changes in land-water interactions; and, finally, and (5) collecting more data to implement a rigorous calibration and validation of hydrological, water quality and ecological parameters. All these aspects may provide a useful contribution to increase the availability of what will become a commonplace management and planning tool for ecological conservation and adaptation policies of sensitive aquatic ecosystems such as those in coastal areas.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.02.131.

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